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ASSESSMENT OF POTENTIAL BIOLOGICAL HAZARDS FROM PROJECT RULISON

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Preface

This analysis of the radiological hazards of Project Rulison was undertaken at the informal request of members of the Nevada Operations Office of the U.S. Atomic Energy Commission.

The meteorological source terms for much of the analysis were supplied by the Environmental Sciences Service Administration, Air Resources Laboratory, Las Vegas. The total radionuclide inventory was taken from References 2 and 3.

Our analysis is incomplete in that we have not considered all possible routes of entry of radionuclides to man. Other groups have been requested to perform analyses of other possible routes in addition to those considered here. Questions relating to hydrology have been investigated by the U.S. Geological Survey and by Isotopes, Inc. Hazards via some food-chain routes not considered in the present analysis have been evaluated by the Battelle Memorial Institute, Columbus. Many additional evaluations have also been performed by the U.S. Public Health Service Southwestern Radiological Health Laboratory and the Nevada Operations Office.

It should be understood that this analysis is therefore not a complete analysis of all radiological hazards, nor was it intended to be. Several specific questions are addressed in the analysis; the absence of other considerations should not be construed as meaning that we felt them to be unimportant, but that they were being considered in detail by others.

Contents

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Abstract

Project Rulison, an underground nuclear detonation designed for the stimulation of natural gas reservoirs, was detonated on 10 September 1969. This report evaluates the off-site radiological hazards associated with the proposed reentry and testing operations. When flaring of the natural gas is begun the major isotope to be released will be tritium (^3H). The report therefore considers several questions concerning the biology and physics of tritium in the light of the data available in the literature. The general conclusions are that ordinary calculations of absorbed dose based on energy uniformly distributed throughout a volume are sufficient to describe the observed effects. No special mechanisms need be considered to account for the extent of the effects observed. Also, when animals are constantly exposed to ^3H, their organic molecules, especially their DNA, have ^3H-^1H ratios essentially the same as that of the body water. Thus, it appears that there is no increase in the incorporation of tritium into DNA as a result of the ingestion of tritiated DNA by ascending trophic levels. The concentration of ^3H in the environment seems to be the controlling factor.

Dose estimates to the surrounding population from inhalation and food-chain pathways were also made from source terms predicted for the Project Rulison operation. From these dose estimates, the risks are calculated for genetic loss, leukemia, cancer and nonspecific life-shortening for the individuals living in the vicinity of the Rulison site. If the maximum average estimated dose via the food chain is taken to be 20 mrem (a dose which can be reduced by employing appropriate precautions), the total detriment to the first generation (genetic loss plus nonspecific life-shortening) represents a possible increase of 0.007% over the natural incidence of genetic loss and malignancy. The increase could be zero. In the interest of public safety, these source terms, dose estimates and risk calculations are made in a very conservative fashion such that upper-limit risks are reported.

Introduction

Project Rulison,[1] the second experiment in the stimulation of natural gas reservoirs by nuclear explosives, was detonated on 10 September 1969. It is presently planned[2,3] to reenter the cavity six months after the shot and at the same time to begin flow testing to determine the cavity volume produced by the explosive and the rate at which natural gas will flow from the "stimulated" reservoir.[2] During

Page 2 is missing from this document.

1.4 to 1.5 times higher than that predicted by the ICRP model.

Somewhat similar results were reported by Koranda et al.[8] They analyzed tritium in the body water and the lyophilized organ residues of 95 kangaroo rats that lived for several generations in the elevated tritium environment surrounding the Sedan crater at the Nevada Test Site. They observed that the tritium activity per gram of hydrogen in the lyophilized residues of six organs averaged 1.5 times as high as that in the body water. They concluded after extrapolating their data to man in a situation of continuous low-level exposure that the body burden and the dose would be 1.8 times that predicted by the ICRP model.

These results, however, are in marked contradiction to those reported by Thompson and Ballou.[9] They exposed mature female rats to constant levels of tritium oxide, mated them after six weeks of exposure, maintained the exposure during the intrauterine development and the nursing of the offspring, and then maintained the offspring on the same exposure level for six months. At that time the offspring were removed from exposure to tritiated water; they were sacrificed at intervals, and various tissues were analyzed to determine the tritium content of their combustion water. In those animals killed immediately after cessation of exposure, most of the tissues yielded combustion water whose tritium concentration was 20 to 30% of the concentration in the body water during exposure. They interpreteted this to mean that 20 to 30% of the organically bound hydrogen (both "freely exchangeable" and "firmly bound") was derived from body water.

If we accept this interpretation as correct, which seems reasonable from considerations of biochemical pathways of hydrogen incorporation,[10] we must assume that the deer studied by Evans[7] must have received most of the tritium in their tritiated organic molecules from the tritiated organic molecules in their food supply and not from their body water. This again seems reasonable since the exposure to tritium was continuous and would have provided adequate opportunity for the food sources utilized by the deer to be contaminated and the plant organic material to be labeled by tritium.

The kangaroo rat utilized in the studies of Koranda et al.[8] is a unique animal which can subsist on dry seeds alone, without any free water. The exposed animals, then, derived most or all of their tritium from the ingestion of tritiated organic materials, and the tritium in their body water came mainly from the catabolism of tritiated organic materials. These rats would exchange tritium between their body water and atmospheric water of lower tritium concentration and would also occasionally drink free water deposited by rain. Under these unique circumstances it seems reasonable that the body water should be lower in tritium than the combustion water.

We can conclude, then, that neither the study of Evans nor that of Koranda et al. can be interpreted as a valid argument for the reduction of the ICRP-recommended MPCs, which are intended solely for either the breathing of air or the ingestion of water contaminated with tritium oxide. Clearly, they were not intended to apply to situations in which the organic portion of the food supplies is

concomitantly contaminated.[4] There remains, however, the fact that some organically bound hydrogen is derived from body water, and that this organically bound reservoir of hydrogen atoms (or tritium atoms) has components that turn over slowly compared to body water.[8,9,11]

For purposes of evaluating continuous exposure to tritium, it might therefore appear appropriate to lower the MPC values by 10 to 20% to account for the additional body burden accumulated in the organic compartment. However, in comparison with the overall uncertainties involved in the establishment of MPCs which are given to only one significant figure, this would not appear to be useful or meaningful.

In situations involving acute exposure to tritium, somewhat different conclusions might be reached due to the long turnover times observed for the organically bound components. Even single large doses of tritiated water are effective in labeling the dry tissue solids of mice. One day after such an administration, Siri and Evers[12] observed that the tritium in dry tissue solids amounted to 7% of the injected dose with about 1.5% of the total firmly bound by metabolic processes. Some human data are also pertinent to this problem. Snyder et al.[13] followed the tritium level in an individual accidentally exposed to tritiated water vapor, and observed that body-water elimination followed a curve described by the sum of two exponentials with half-lives of 8.7 and 34 days. The total tritium in the second component appears to have been about 0.5% of that in the first component, and it contributed about 2% of the total dose. Reinig and Sanders[14] studied another case of accidental inhalation, in which the body burden, 46 mCi, was large enough so that the tritium excretion could be followed for 415 days. The observed urinary excretion pattern was assumed to indicate that the tritium was distributed among three compartments with half-lives of 6.14, 23.4, and 344 days. The highest percentages of assimilated tritium in the longer half-life compartments were 0.54% and 0.26%, respectively. Reinig and Sanders made conservative estimates of the dose contributions of the tritium in each of these compartments, and concluded that the rapidly turning-over compartment (body water) gave a maximum dose commitment of 3.52 rem whereas the slower compartments gave 1.37 rem. It seems apparent, therefore, that the dose calculated on the basis of distribution of tritium solely within the body water should be increased by 40%. However, if we calculate the infinite dose using the ICRP model with a body water half-time of 12 days, the result is 5.8 rem.* This discrepancy is due to the conservatism incorporated into the ICRP model regarding the half-time of body water. Whereas the ICRP uses the value of 12 days, the two experiments mentioned above actually yielded considerably shorter values, and in general the half-times measured by many experimenters average 9.5 days[16,17] rather than 12 days.

We can therefore conclude that the values obtained with the ICRP model do

*Reinig and Sanders used a relative biological effectiveness (RBE) of 1.7 in their calculations. This value has subsequently been changed by the ICRP to 1.0,[15] but 1.7 was used in this calculation so it could be compared with the values of Reinig and Sanders.

not significantly underestimate the doses delivered by either chronic or acute exposures to tritiated water by ingestion or to tritiated water vapor by inhalation and skin absorption.

INCORPORATION OF ^{3}H INTO DNA

The second question relates specifically to the incorporation of tritium into DNA and other organic components of long turnover times. If such compounds become tritiated in low trophic levels of an ecosystem and are subsequently passed upward through a food chain, the possibility has been raised that the ultimate exposure to man eating such foodstuffs could be substantially higher than that predicted from the average concentration of tritium released to the environment.[18]

We must first consider whether tritium as tritiated oxide can be incorporated into DNA. Considerations of the biochemical pathways that generate nucleic acids clearly indicate that tritium in body water is incorporated into DNA,[19] although experimental data on this point are lacking. Relevant data are those of Metzger et al.,[20] who injected rats intraperitoneally with tritiated water (25 μCi/g body weight), sacrificed them four hours later, and then assayed the firmly bound tritium activity in macromolecular fractions of the liver. They observed approximately equal tritium concentrations (70 cpm/mg) in nucleic acids and proteins. The efficiency cited for their tritium determination was 20%. Using this figure, we calculate that the specific activity was 0.16 μCi/g; the concentration in the body water would have been about 40 μCi/g. Although the fractional activity is small, it is clear that even brief exposures to high concentrations of HTO are effective in incorporating tritium into rat liver DNA and proteins, and that the levels of incorporation in the two classes of macromolecules are not essentially different. It also seems reasonable to assume that most biological organisms would have rather similar characteristics.

We would next like to consider to what extent the label in the DNA contained in an initial food link might be passed on to DNA in subsequent links. Unfortunately, there appear to be no data available to make such analyses directly. While no studies are available on the transfer of tritiated nucleic acids through ascending trophic levels, several studies have been done on the incorporation of tritiated DNA precursors into newly synthesized DNA.

Hinrichs et al.[21] injected tritiated thymidine (^{3}HTdR) intraperitoneally into mice and determined the percentage of injected label that was incorporated into total-body DNA. They observed that with doses less than 30 μCi per mouse, less than 6% of the injected radioactivity was incorporated into the total DNA. Most of the injected activity was rapidly catabolized to HTO, as has also been observed in humans following intravenous injection of tritiated thymidine.[22]

Lambert and Clifton,[23] comparing the efficiency of incorporation of tritiated thymidine into rat DNA by two routes, intraperitoneal injection and ingestion, observed that ingestion was only one-eighth as efficient as intraperitoneal injection. From the two experiments, we can conclude that after the oral ingestion of tritiated thymidine only 1% of the label is incorporated into DNA, and, for mammals at least, we would expect little

transference of tritium contained in ingested DNA into newly synthesized DNA.

The weight ratio of DNA to body water has been estimated to be about 10^{-3} for both mouse[21] and man.[24] If similar ratios pertain to all plant and animal species, this would further argue that most of the tritium incorporated into newly synthesized DNA would come from sources and pathways other than the direct incorporation of tritium contained in ingested DNA. Additionally, there is no assurance that the ingestion of DNA precursors represents an adequate model for the ingestion of food containing intact DNA. Digestive processes may not provide precursors that can be readily utilized.

Relevant experimental data are those of Hatch et al.[25] on the tritium activity in the liver DNA of kangaroo rats collected at Sedan Crater. They observed the same tritium activity per gram hydrogen in the liver DNA as in lyophilized whole liver tissue. This indicates that tritium was not preferentially incorporated into their DNA relative to their whole liver tissue, even though their tritium exposure was derived mainly from the ingestion of tritiated organic compounds.

The available evidence, then, gives us no reason to believe that an organism of any trophic level, continuously exposed to relatively constant levels of tritium, would accumulate significantly higher levels in its long-lived macromolecular components than in its other organic compounds or its body water. (The kangaroo rat or any organism whose hydrogen intake is derived mainly from organic matter would be an exception to this, as far as body water is concerned.) Patterns of incorporation after acute exposures (or after the termination of continuous exposures) are clearly different; the relative concentrations would be controlled by the relative turnover rates of the various organic and body-water components. We have already discussed evidence indicating that after transient exposure, the long-lived organic components will eventually contain higher concentrations of tritium than the shorter-lived components.

It should be kept in mind that these conclusions are based upon rather limited experimental data. However, the extrapolation of the available data to the Rulison situation can be made with reasonable confidence, since the organisms of primary concern are mammalian, and the food chains are not complex. We feel much less confident about extrapolating these results to all members of complex (for example, aquatic) ecosystems involving a great diversity of organisms within food chains. Isotopic fractionation mechanisms for the hydrogen isotopes are known. Man, for example, exhales water vapor containing 94% of the tritium concentration in plasma,[26] and the pigeon has been reported to be unusually efficient in this process, with its expired water vapor tritium level only about 50% of that of the urine or the blood.[12] Other fractionation effects observed between deuterium and protium have been summarized by Bowen.[27] We do not know how widespread these processes are, and there appear to be no sufficiently detailed studies of the behavior of tritium in complex ecosystems to permit a full evaluation of the presence of these processes.

ENERGY DISTRIBUTION FROM ^3H DECAY

Question three, dealing with the dose estimates and subsequent effects attributable to the low-energy tritium β-particle, has been examined in considerable detail by Bond and Feinendegen[28] and by Feinendegen,[11] on the basis of their own work and the available literature. Some of the early somatic effects observed after tritium exposure and compared with those after x-irradiation are cell killing in tissue culture,[29-33] cell killing in intact mammals,[28,34,35] mitotic delay,[28,32] and various types of cytogenetic abnormalities.[36,37] Late-term somatic effects which have been studied include carcinogenesis.[38] Genetic effects include both dominant lethal induction[39,40] and point mutations in the form of recessive lethals.[41-43]

Bond and Feinendegen[28] concluded that all of the early somatic effects resulting from tritium exposure could be accounted for on the basis of the dose absorbed by the cell nucleus. The extent of the effect for any given absorbed dose as a result of tritium exposure was quantitatively similar to that observed following an equivalent absorbed dose of x-irradiation. In other words, no special calculations need be considered other than the absorbed dose due to the random disintegration of ^3H atoms, regardless of whether the ^3H atom is in the form of HTO or actually incorporated in DNA as ^{3}HTdR. The same quantitative correlation between absorbed tritium dose and absorbed x-ray dose also appears to occur for the long-term somatic effects and the genetic effects. In some cases the absorbed dose from ^{3}HTdR or HTO appears to be slightly less effective than an equivalent dose from x-irradiation while in other cases it appears to be slightly more effective. However, if we allow for experimental error and for the technical difficulties in such experiments, it appears that the dose absorbed by the cell nucleus as a result of tritium decay accounts quantitatively for the amount of the response, and it does not exceed that expected from a similar dose of x-rays.

Lambert[34] recently published results of experiments (not included in the analyses of Bond and Feinendegen) designed to look at the death of intermediate and type B spermatogonia after irradiation of mouse testes. The testes were irradiated externally with x-rays or internally with ^{3}HTdR or HTO. Again, the effects were quantitatively quite similar if the absorbed dose from tritium was compared to a comparable absorbed dose from x-irradiation. Tritium was slightly more effective than x-rays in these experiments and HTO was more effective than ^{3}HTdR. These results are similar to those of Johnson and Cronkite,[35] in which ^{3}HTdR and ^{60}Co γ-rays were compared for effectiveness in the induction of spermatogonial killing.

Various experiments have been done to compare tritium β-irradiation to ^{60}Co γ-irradiation or x-irradiation with respect to their effectiveness for producing a given biological effect. In some experiments the tritium β seems to be slightly less effective for an equivalent dose while in others it seems to be slightly more effective. In some experiments where RBE values of 1.6 to 2.0 are calculated, the uncertainties in the dose calculations make the results very tenuous, and the

results should be regarded with circumspection.

Bond and Feinendegen[28] have pointed out that RBEs of 1.3 or 1.4 for tritium β-irradiation relative to ^{60}Co γ-irradiation are closer to unity when compared to 250 kVp x-rays. Also, they have pointed out that RBEs of 1.3 to 1.7, which have been observed for a variety of endpoints in intact mammals are based on calculations which assume uniform distribution in the body water and uniform whole-body dose. However, a large part of an animal is bone and this component has a minimal uptake of HTO. When one takes into account this inhomogeneity in the distribution of tritium then the RBE value is nearly unity.

Recently the ICRP[15] adopted a quality factor (or RBE) of 1.0 rather than 1.7 for β energies less than 0.03 MeV; therefore, the quality factor for tritium is taken to be unity, and our dose calculations are based on this value.

TRANSMUTATION OF ^{3}H

Question four concerns the possible effects of transmutation of DNA as a result of ^{3}H decay. Bond and Feinendegen[28] also considered this problem. Their general conclusion is that no experiments have shown that transmutation effects have any role either in cell lethality (where results can be accounted for entirely by the absorbed doses) or in other cell damage such as cytogenetic abnormalities (where there is no correlation between the site of the chromosome break and the site of incorporation of the label).[37] Rachmeler and Pardee[43] have, however, apparently shown a transmutation effect in the induction of mutants in bacteria after labeling with ^{3}HTdR. Bond and Feinendegen stated that transmutation might be the explanation of the data of Kaplan et al.[41] who looked for sex-linked recessive lethals in Drosophila after labeling with ^{3}HTdR. The distribution of lethal mutations along the x-chromosome was nonrandom and different from that observed after x-ray exposure. However, in a more recent experiment Kaplan et al.[42] found the distribution of mutations along the x-chromosomes after labeling with tritiated deoxycytidine (^{3}HCdR) to differ slightly from that observed after ^{3}HTdR or x-rays. When the distributions resulting from the two different labels (i.e., ^{3}HTdR and ^{3}HCdR) were summed, the results were like those after x-rays. This was interpreted as suggesting (1) that the combined distribution reflects variations in the regional content of DNA along the x-chromosome and (2) that the difference in the distributions of lethal mutations from ^{3}HTdR and from ^{3}HCdR demonstrates variations in thymine and cytosine content within the chromosomal DNA. Therefore, it is not necessary to invoke transmutation effects to explain these results.

Exposure to tritiated water rather than ^{3}HTdR results in randomly labeled organic molecules; the effect of transmutation would be negligible, since tritium would not be preferentially incorporated into DNA and since DNA comprises only about 0.1% of the organic body weight.

One additional problem should be considered: the unique situation of the fetus continually exposed to tritium throughout fetal life. The experiments of Khan and Wilson[44] and Thompson and Ballou[9]

indicate that fetal exposure of rats leads to the incorporation of tritium into organic molecules throughout organogenesis with subsequent long half-times depending on the site of incorporation. Also, as stated by the ICRP,[45] several experiments have suggested that the effect per rad on the induction of neoplasia following irradiation of the fetus is greater than the effect on irradiated children and adults by a factor between 2 and 10. Miller[46] has recently questioned the validity of such a conclusion but has not presented any compelling evidence that it is incorrect.

These considerations suggest that the critical segment of the population at risk would be the fetus. If the fetus is indeed ten times as sensitive as the adult, one could argue that dose standards should be reduced by an order of magnitude when applied to the exposure of pregnant women. The effect of tritium incorporation into organic molecules is much more difficult to assess, in that it is not certain what percentage of the tritium under such conditions is bound to the various types of organic molecules. Obviously, some are much more critical than others.

Estimated Doses from Tritium

In the Rulison basin the wind direction varies from daytime to nighttime. Nighttime flow is in a northerly direction down the valley; daytime flow is up the valley in the opposite direction. In the daytime the flow eventually reaches an altitude such that it moves with the prevailing northeasterly winds. Therefore, the predicted source terms, which we use here, include air concentrations and ground depositions for a 12-hr daytime period and a 12-hr nighttime period for both the accident situation and the normal flaring operation.[2]

The accident situation assumes that there would be no control over the gas flow at the wellhead and that uncontrolled blowout would occur through an open drill hole. The source terms for the accident case are calculated assuming that under such conditions 100% of the tritium present at 180 days after detonation would be released in a 24-hr period.

The normal flaring operation consists of controlled venting of the cavity volume. Source terms for tritium for the flaring case were developed assuming that 24% of the total ^{3}H activity will be vented in the first 3-day flaring operation.[2,3] The predicted air concentrations and ground depositions from which the doses are calculated assume that the cloud travels in a straight line—that is, that there is no meandering of the plume. Actually, one would expect the plume to meander during a 12-hr period. Therefore, because of the different wind directions during daytime and nighttime and the probable meandering of the plume, one would not expect any one person to be in the cloud for any 24-hr period, and summing the dose over an entire day (or over a 3-day flaring operation), would certainly produce a maximum dose estimate. Releases after the first 3-day flare will involve smaller quantities of activity and will be spread over many months. Therefore, such variables as plume meandering and wind direction will have a much greater effect, and the dose

received will be much less than that from the initial flare.

The source terms used in the analyses of the accidental release and the normal flaring operation were taken from Refs. 2 and 3. It should be emphasized that these source terms assume that all of the tritium produced will be in the gaseous phase. It is expected, however, that a major fraction of the tritium will be contained in water in the cavity. Detailed studies of the gas produced by the first gas-stimulation experiment, Project Gasbuggy, indicate that only 5% of the total tritium was actually in the gaseous phase.[47] Fleming[48] has concluded that the source terms in Refs. 2 and 3 overestimate the tritium concentration in the Rulison gas by an order of magnitude. This conclusion was based upon predictions of the chemical reactions thought to have occurred within the Rulison cavity and upon the analysis of the initial gas sample drawn from the Rulison wellhead. We chose to use the more conservative values in our analyses because predictive experience is very limited and the initial sample may not be representative of the actual cavity gas. Therefore the exposure estimates for the first 3-day flare have been determined in a very conservative fashion and are probably reasonable estimates of the total dose that an individual might possibly receive from the total flaring operation.

Knox[49] has supplied source terms for self-induced rainout, a phenomenon that may occur at or near the wellhead under winter weather conditions. It consists of the condensation of water vapor out of a supersaturated gas cloud as it leaves the flaring stack.

Dose calculations will be made for the accident case and the normal flaring operation considering both air inhalation and ingestion via the food chain. The approach of Ng et al.[50] for determining unit-rad depositions* was used for the dose estimates via the forage-cow-milk pathway. The unit-rad deposition value for tritium via the forage-cow-milk pathway is 99 $\mu Ci/m^2$.

The unit-rad deposition also was calculated for the soil-root pathway using the technique of Ng et al.[50] However, because of recent information concerning tritium movement in various ecosystems, we have made a few numerical changes in some of the constant parameters in the basic equations.[51] The approach used by Ng et al. assumes that tritium stays within the top 20 cm of the soil surface after deposition. Koranda and Martin[52] showed recently, however, that the peak concentration of tritium moves up and down in the soil and reaches depths down to 7 or 8 ft. Therefore, we have assumed that, on the average, tritium is distributed in the top 100 cm of soil.

It was also assumed in the original analysis that the half-residence time for tritium in the soil was equal to its radioactive half-life of 12.3 years. However, recent evidence has shown the half-residence time for tritium to be about 29 days[53] in a tropical rain forest and about 18 months[52] in the dry desert region of the Nevada Test Site. Most areas would probably fall somewhere between these two extremes. In our

*The unit-rad deposition is the amount of radionuclide in $\mu Ci/m^2$ from a single contaminating event that will result in a 30-yr dose of one rad.

analysis we used the longer of the observed half-residence times, i.e., 18 months.

With these two major changes in the equations used by Ng et al., We found the unit-rad deposition for ^3H to be 1600μCi/m^2. Therefore, the dose received via the soil-root pathway would be at least 16 times less than that received via the forage-cow-milk pathway (99 μCi/m^2 unit-rad deposition).

The level of exposure via the forage-cow-milk pathway will be affected to some degree by the time of year when the flaring operation is conducted. If there is still any considerable amount of snow on the ground, the cows will not be on pasture and much less tritium would remain on the plants than would be deposited on them if the flaring were conducted after the snow melt. Also, the final tritium concentration in soil water would be less if flaring were conducted while there was still snow cover on the ground.

Another possible source of contamination that we have not considered is the contamination of water supplies. However, due to the large dilution factors involved and the ease of monitoring such supplies in the Rulison vicinity, the dose from this source should be negligible. An evaluation of this problem has been published.[2,3]

ACCIDENTAL RELEASE

Inhalation

Our first calculation is for air inhalation at a distance of 5 km (distance from wellhead to nearest population) as a result of a daytime accident. At an effective release height of 300 m, which gives the highest expected concentrations, the predicted tritium air concentration is 3.5×10^{-8} Ci/m^3. If we use the ICRP value for the volume of air breathed by standard man, i.e., 2×10^7 cm^3 per day or 10^7 cm^3 per 12 hr, then the number of microcuries inhaled in a 12-hr period is 0.35 μCi. The amount inhaled is essentially all absorbed.[54] For a 70-kg man this results in a first-year dose* of 3×10^{-2} mrem, assuming a 12-day half-life for tritium in the body and a relative biological effectiveness (RBE) of 1.0.[15] The nighttime air concentration at 5 km as a result of an accident situation is less by about a factor of 1000 than the daytime case and would, therefore, give a dose of 3×10^{-5} mrem for a 12-hr release period. At 15 km the nighttime release would lead to a dose of 2×10^{-3} mrem while the daytime release would lead to a dose of 7×10^{-3} mrem.

Dry Deposition

The amount of tritium deposited on the ground at 5 km as a result of an accident is 1.5×10^{-5} Ci/m^2 for the daytime case.[2] The unit-rad deposition value for the forage-cow-milk pathway for tritium according to Ng et al.[50] is 99 μCi/m^2. This means that a child drinking one liter of milk per day from cows grazing on pasture initially contaminated with ^3H at 99 μCi/m^2 would receive a dose of one rem.† The daytime deposition at 5 km therefore results in a dose of about 0.2 rem. The nighttime deposition at 5 km is less than this by a factor of 1000. At 15 km the nighttime and daytime depositions would lead to doses of 10 mrem

*First-year dose and 30-year dose are the same for an acute exposure to tritium.

†The unit-rad deposition value for the forage-cow-milk pathway was derived with the assumption that the half-residence time of tritium on forage is 14 days.

and 40 mrem, respectively. If the entire chimney were emptied in a 24-hr period in the same location, these two values would have to be summed. As was pointed out previously, the unit-rad deposition via the soil-root pathway is more than 16 times higher than that via the forage-cow-milk pathway; therefore, the dose received via the soil-root system would be at least 16 times less than the doses calculated for the forage-cow-milk pathway.

FLARING OPERATION

Inhalation

For the flaring operation the initial calculation for the dose due to air inhalation is made for the daytime situation at 5 km. This is the maximum value predicted for air concentrations as a result of flaring and is, therefore, the limiting case for the air inhalation situation. The predicted air concentration is 2.6×10^{-9} Ci/m^3. Again assuming that the inhaled air volume for a 12-hr period is 10^7 cm^3, the short-term body burden is 0.026 μCi. This is less by more than a factor of 100 than the accident case; hence, the dose would be 2×10^{-3} mrem. The dose for the nighttime release case at 5 km is less than this by a factor of 500. At 15 km, the 12-hr daytime flaring would lead to a dose of 5×10^{-4} mrem, and the 12-hr nighttime flaring would lead to an inhalation dose of 2×10^{-4} mrem.

The total inhaled dose at 5 km as a result of a 3-day flaring period is then approximately 6×10^{-3} mrem. The total inhaled dose at 15 km would be less by a factor of about three than at 5 km, i.e., it would be about 2×10^{-3} mrem.

Dry Deposition

The deposition of tritium onto plants as a result of the flaring operation can be calculated using the source term value of 10^{-4} Ci-sec/m^3 for the integrated activity at 5 km for daytime release, and a value of 10^{-2} m/sec for the deposition velocity.[2] The result is 10^{-6} Ci/m^2. This is less by a factor of 100 than the deposition for the maximum accident case and leads to a dose of 10 mrem for each 12-hr period for the forage-cow-milk pathway. Nighttime deposition is less by a factor of 1000 and therefore leads to a dose of 10^{-5} rem for each 12-hr period. The total dose as a result of the 3-day flare would then be about 30 mrem. At 15 km, the deposition from daytime release leads to a dose of 3 mrem for each 12-hr period; nighttime deposition leads to a dose of 1 mrem. The total dose resulting from a 3-day flare is therefore approximately 10 mrem. The doses resulting from the soil-root pathway would again be less by a factor of 16 than the above estimates for the forage-cow-milk pathway.

RAINOUT

Tritium can also be deposited on the ground surface by the process in which natural rain brings down the activity contained in the plume.

For the accident case, it has been estimated[2] that the specific activity of this water could be between 1.8×10^{-3} and 4.8×10^{-3} μCi/g; this can be compared to the RCG value of 1×10^{-3} μCi/g for drinking water. It would take about 2 cm of rain to bring down enough activity to equal the unit-rad deposition for the forage-cow-milk pathway. This amount of water, however, could not possibly be retained by

the plant surfaces and would move into the soil water. Dose contributions would therefore be reduced to less than the dose calculated to arise from dry deposition and entry into the forage-cow-milk pathway. The same conclusions would also apply to the flaring operation.

SELF-INDUCED RAINOUT

Knox[49] has estimated self-induced rainout under winter weather conditions at a distance of 2 km. His value, 30 g/m^2-hr for each 12-hr period, gives 720 g/m^2. The activity of the water is 10^{-7} Ci/g or, therefore, 7.2×10^{-5} Ci/m^2. If 25% of this self-induced rainout remains on the plants, then there would be essentially 1.8×10^{-5} Ci/m^2, which would lead to a dose through the forage-cow-milk pathway of 0.2 rem for each 12-hr period. Knox estimates that deposition in the accident case would be about five times

Table 1. Calculated whole-body dose for inhalation and ingestion of tritium.[a]

	Distance from source (km)	Accidental		Operational (flaring)		
		Daytime (rem)	Nighttime (rem)	Daytime (rem)	Nighttime (rem)	Total, 3-day operation[b] (rem)
Air inhalation[c,d] (12 hr at highest concentration)	5	3×10^{-5}	3×10^{-8}	2×10^{-6}	4×10^{-9}	$\sim 6 \times 10^{-6}$
	15	7×10^{-6}	2×10^{-6}	5×10^{-7}	2×10^{-7}	$\sim 2 \times 10^{-6}$
Dry deposition[d]						
Via forage-cow-milk food chain[e]	5	2×10^{-1}	2×10^{-4}	1×10^{-2}	1×10^{-5}	$\sim 3 \times 10^{-2}$
	15	4×10^{-2}	1×10^{-2}	3×10^{-3}	1×10^{-3}	$\sim 1 \times 10^{-2}$
Via soil-root food chain[f]	5	1×10^{-2}	1×10^{-5}	6×10^{-4}	6×10^{-7}	$\sim 2 \times 10^{-3}$
	15	3×10^{-3}	6×10^{-4}	2×10^{-4}	6×10^{-5}	$\sim 6 \times 10^{-4}$
Self-induced rainout[g] (via forage-cow-milk food chain)[e]	<2	1	1	2×10^{-1}	2×10^{-1}	~1

[a] It has recently been estimated[48] from an analysis of a sample of gas from the Rulison wellhead that the amount of tritium present in the gas phase is less by nearly a factor of 10 than that initially assumed in Refs. 2 and 3 and used for estimating the air concentrations and ground depositions. Our dose esimates are based upon the air concentrations and ground depositions in Refs. 2 and 3, and if the amount of tritium available is less by a factor of 10, then the above dose estimates would also be reduced by a factor of 10. The maximum expected dose would therefore be 3 mrem (excluding self-induced rainout; see footnote g).

[b] The exposure estimates for the first 3-day flare have been determined in a very conservative fashion and are probably reasonable estimates of the total dose that an individual might possibly receive from the total flaring operation.

[c] Standard man inhales 2×10^7 cm^3 per day (ICRP).[4]

[d] Source terms from Refs. 2 and 3.

[e] Dose to child via forage-cow-milk pathway from Ng et al.[50]

[f] Dose to child via soil-root pathway from Burton[51] and Ng et al.[50] with modification (see text).

[g] From Knox.[49] Self-induced rainout might be expected only during winter weather conditions. There are neither people nor dairy cows within 2 km of the wellhead; hence no actual dose contribution via this route would be expected.

greater; the dose would be about 1 rem for each 12-hr period. However, there are no people and no dairy cattle within 2 km of the wellhead,[2] and self-induced rainout is not expected to occur at greater distances.

The results for the various situations are summarized in Table 1. The significant point to be made from these data is that the food pathway could contribute by far the greatest percentage of the dose. The dose that could be contributed by air inhalation over a 12-hr period is trivial compared to the dose that could be received via the forage-cow-milk pathway as a result of such air concentrations.

Hazards from Other Radionuclides

GASEOUS ACTIVITIES

In addition to tritium, other radionuclides are present in the chimney in a gaseous form and are expected to be released by the flaring operation or in the event of an accident. The radionuclides of possible concern are ^{14}C, ^{37}Ar, ^{39}Ar, ^{85}Kr, and ^{133}Xe, although only ^{85}Kr is present in significant quantity. Total inventory values and predicted air activities are available.[2]

For the gases, the ICRP model considers hazards arising from submersion in a cloud of the gas. Calculations using this model indicate that the only gaseous radionuclide of significance is ^{85}Kr, which would contribute a dose equal to 25% of the dose due to inhalation of tritium. The ^{14}C could also be inhaled, but would not contribute any significant dose by this route due to the relatively very small amount of this radionuclide present within the chimney.

Dry deposition values have also been given[2] for ^{14}C. Dose contributions from this radionuclide via the forage-cow-milk and the soil-root pathways were evaluated using the model of Ng et al. and are less by orders of magnitude than the dose contribution from tritium.

Ng et al.[50] published unit-rad deposition values for the noble gases for both the forage-cow-milk and the soil-root pathways. However, these values cannot be used in a meaningful fashion because deposition velocities for noble gases are not known. As an alternative approach, we started with the maximum air concentrations given for the noble gases,[2] and assumed that the gases can be deposited only by being absorbed by water. The maximum concentration of the noble gases in water was calculated using Henry's Law, assuming instantaneous equilibrium. This concentration of noble gases in water was then assumed to be directly transferred to man and eliminated only by radioactive decay. These extremely conservative assumptions indicated that ^{85}Kr was the noble gass radionuclide of most concern, but that the dose calculated in this manner was less by a factor of 10 than that calculated from the ICRP model of submersion in the original air concentration of ^{85}Kr. Therefore, any possible dose contribution by radionuclides of noble gases via food-chain pathways is insignificant.

PARTICULATE ACTIVITIES

The hazards to man via the forage-cow-milk and the soil-root pathways for

all individual particulate radionuclides in the chimney can also be estimated, using the conservative values of Ng et al.,[50] and compared to the tritium hazard.

This analysis is made by considering the total chimney inventory at six months after detonation and the critical organ unit-rad deposition values. The more conservative values for the infant are used. Dividing the inventory (in μCi) by the unit-rad deposition value gives the m^2-rad value, which, when multiplied by the fractional deposition per square meter, gives the dose in rads. More importantly, the m^2-rad value allows one to compare directly the relative risk for different radionuclides.

This done using inventory values given in Ref. 2; the results indicate that the radionuclides of most concern are ^{90}Sr, ^{106}Ru, ^{95}Nb, ^{89}Sr, and ^{137}Cs. For the most hazardous radionuclide, ^{90}Sr, we can say that if its fractional deposition (or its air concentration) is 10^{-4} of that of tritium, it would constitute a hazard equal to that of tritium. This same estimate of relative hazard for ^{90}Sr can be derived also by considering only the inventory relative to the RCG values for air concentration. However, essentially none of these isotopes is expected to escape from the cavity.[2, 3]

Risk Estimates

Our dose estimates are derived from the source terms for the normal flaring operation, since it is the most likely situation. The largest estimated dose to the population results via the foodchain pathways. However, this is a controllable situation such that any dose via these routes (forage-cow-milk and soil-root) would be reduced if necessary by appropriate Public Health measures. With this in mind, we will present risk estimates for doses received through the forage-cow-milk pathway, and if the primary exposure is by air inhalation, then the exposure is lower by a factor of 2000. The largest dose via the forage-cow-milk pathway due to dry deposition is 30 mrem at 5 km, while at 15 km the dose is 10 mrem. Therefore, for persons within 15 km of the Rulison site, the average upper-limit dose is approximately 20 mrem. The largest total dose due to inhalation during a 3-day flaring operation is 0.01 mrem at 5 km.

We established earlier in this report that the average absorbed dose resulting from tritium exposure can be used satisfactorily to make such calculations. The details concerning the risk factors used to derive the risk estimates summarized in Table 2 are given in the Appendix. The results are summarized in Table 2. These are conservative upper-limit estimates based on extrapolation from high doses and high dose rates. We have not taken into account any possible dose-rate effects or possible repair mechanisms at these lower doses, although various studies indicate that such processes might operate in mammalian systems. For example, radiation-induced mutation frequencies show a considerable dose-rate effect, with acute exposure resulting in

Table 2. Summary of the calculated upper-limit risk per person due to exposure to 20 mrem in one year.[a,b,c]

	Increased incidence per person due to radiation		Natural incident per person
	Lower limit	Upper limit	
First generation			
Genetic loss[d]	0	4×10^{-6}	2×10^{-1}
Adult leukemia	0	4×10^{-7}	5×10^{-3}
Childhood leukemia	0	1×10^{-7}	2×10^{-5}
Other adult malignancy	0	8×10^{-7}	1×10^{-1}
Other childhood malignancy	0	3×10^{-7}	2×10^{-5}
Nonspecific life-shortening[e]	0	1×10^{-5}	
Total for first-generation (nonspecific life-shortening plus genetic loss)	0	$\sim 2 \times 10^{-5}$	
Over all times			
Additional genetic loss	0	1×10^{-4}	
Total detriment over all time	0	$\sim 2 \times 10^{-4}$	

[a]It has recently been estimated[48] from an analysis of a sample of gas at the Rulison wellhead that the amount of tritium present in the gas phase is less by an order of magnitude than that initially assumed in Refs. 2 and 3 and used for estimating air concentrations and ground depositions. Our risk estimates are based upon the air concentrations and ground depositions in Refs. 2 and 3 and if the amount of tritium available is less by a factor of 10, then the above risk factors would also be reduced by a factor of 10.

[b]If exposure due to the food chain were negligible and the primary exposure were by inhalation, then the dose due to inhalation would be 0.01 mrem, and the above risk factors would be reduced by a factor of 2000.

[c]There are approximately 700 persons within a 15-km radius of the Rulison wellhead and approximately 1000 persons within 24 km. The cattle population (67 within 15 km) is randomly distributed throughout this area. With changing meteorological conditions over periods of hours (especially at greater distances from the wellhead), the dose a person could receive via inhalation or via the forage-cow-milk pathway will vary considerably, with an upper limit of 30 mrem at 5 km (this dose assumes the original air concentration and ground deposition values which, as discussed above, may be lower a factor of ten).

[d]Genetic loss refers to the eventual elimination of a deleterious gene. This would be evidenced by abortion, stillbirth, prereproductive death, early embryonic death, lowered fertility, or sterility.

[e]Nonspecific life-shortening includes losses due to cancer plus all other diseases and physiological processes leading to a shortened life-span.

approximately four times as many mutations as chronic exposure.[55,56] In irradiated cells, either in vivo or in vitro, survival increases when the dose is delivered over a prolonged period or in fractions rather than within one short interval.[33,57] Also, fewer chromosome aberrations are observed after chronic or fractionated exposure than after acute exposure.[58,59] Studies of chronic exposure in relation to cancer induction in humans, however, are nonexistent.

Repair processes have also been demonstrated in various organisms and cell lines after ultraviolet irradiation.[60-62] Unscheduled DNA synthesis in mammalian cells has been reported at x-ray doses of 5000 R[61,63] and a type of "repair replication" after very large doses of irradiation has been observed in HeLa S-3 cells.[64] However, the relationship of this repair replication to cell recovery is still speculative. It has also been established that rejoining of x-ray induced breaks occurs in the DNA of mammalian leukemic cells.[65] The rejoining process seems to be rather radioresistant, but again its relationship to cell recovery has yet to be clearly defined.

Enzymatic rejoining of single-strand breaks in the DNA of bacteria has been reported by two different groups,[66,67] and the excision of thymine dimers and mismatched sequences by DNA polymerase has been observed by Kelly et al.[68] It appears that the enzyme may be able to carry out both excision and polymerization, indicating repair of damaged DNA. The relationship of all the observed repair mechanisms to cell killing, or more importantly, to the induction of various types of cancer and genetic effects, is a major question in radiation biology.

It is possible, therefore, that dose-rate effects and repair processes might, for low-dose exposure, reduce the damage below that predicted by linear extrapolation from high doses and high dose rates. However, to be conservative in these calculations we proceeded on a linear extrapolation hypothesis from the observed effects at higher doses.

When the Appendix was originally written,[69] we felt that the weakest evaluation concerned the estimation of the magnitude of nonspecific life-shortening. Hence, we chose a very conservative estimate of 7% reduction in life span per 100 rad. Andersen and Rosenblatt[70] recently reported the results of a study on female beagles in which the median life span was shortened 6.7% per 100 rads, a value in agreement with our estimate. Storer,[71] however, asserted in a recent publication that 1% per 100 rad is probably a conservative number when extrapolated to exposures occurring at low dose rates.

Another recent analysis[72] indicates that the total increase in all radiogenic cancers (excluding leukemia) may be higher than our estimate by more than an order of magnitude. We are not convinced that the data available at the present time justify such a conclusion. However, even if such an interpretation should be correct, our inclusion of nonspecific life-shortening (which includes losses due to death by cancer as well as all other causes) in the evaluation as lives lost would be adequate to cover such an effect. Therefore, when our estimate for lives lost due to life-shortening is combined with the estimate for genetic loss, the total appears to be a conservative upper-limit value for the overall risk estimate.

References

1. Project Rulison Definition Plan, CER Geonuclear Corp., Las Vegas, Nevada, Rept. (1969).
2. Project Rulison Postshot Activities, U.S. Atomic Energy Commission, Nevada Operations Office, Las Vegas, Nevada, Rept. NVO 61 (Preliminary) (November, 1969) (OUO).
3. Project Rulison Postshot Plans and Evaluations, U.S. Atomic Energy Commission, Nevada Operations Office, Las Vegas, Nevada, Rept. NVO 61 (December, 1969).
4. "Report of ICRP Committee II on Permissible Dose for Internal Radiation (1959)," Health Phys. 3 (1960).
5. Radiation Protection Guidance for Federal Agencies, Federal Radiation Council, Washington, D. C., Rept. No. 1 (1960).
6. "Standards for Radiation Protection," Chapter 0524 in AEC Manual, U.S. Atomic Energy Commission, Washington, D. C. (November 8, 1968).
7. A. G. Evans, "New Dose Estimates from Chronic Tritium Exposures," Health Phys. 16, 57 (1969).
8. J. J. Koranda, J. R. Martin, and R. W. Wikkerink, "Biological Half-Life Studies of Tritium in Animals with a Unique Chronic Exposure at Sedan Crater," paper presented at the Health Physics Society Annual Meeting, Denver, Colorado, June 16-20, 1968. (Abstract: Lawrence Radiation Laboratory, Livermore, Rept. UCRL-70914 Abstract (1969)).
9. R. C. Thompson and J. E. Ballou, "Studies of Metabolic Turnover with Tritium as a Tracer. V. The Predominantly Non-Dynamic State of Body Constituents in the Rat," J. Biol. Chem. 223, 795 (1956).
10. R. T. Taylor, Lawrence Radiation Laboratory, Livermore, private communication (1969).
11. L. E. Feinendegen, Tritium-Labeled Molecules in Biology and Medicine (Academic Press, New York, 1967).
12. W. Siri and J. Evers, "Tritium Exchange in Biological Systems," in Tritium in the Physical and Biological Sciences II, Symposium on the Detection and Use of Tritium in the Physical and Biological Sciences, Vienna, 1961 (International Atomic Energy Agency, Vienna, Austria, 1962), p. 71.
13. W. S. Snyder, B. R. Fish, S. R. Bernard, M. R. Ford, and J. R. Muir, "Urinary Excretion of Tritium Following Exposure of Man to HTO - A Two Exponential Model," Phys. Med. Biol. 13, 547 (1968).
14. W. C. Reinig and S. M. Sanders, Jr., "Assessment of Tritium in Man," in Diagnosis and Treatment of Deposited Radionuclides, Monogr. on Nuclear Medicine and Biology, No. 2, H. A. Kornberg and W. D. Norwood, Eds. (Excerpta Medica, Amsterdam, 1968), pp. 534-542.
15. "Report on the Meeting of the International Commission on Radiological Protection (ICRP), April, 1969," Acta Radiol. 8, 258 (1969).

16. H. L. Butler and J. H. Leroy, "Observation of Biological Half-Life of Tritium," Health Phys. 11, 283 (1965).
17. C. R. Richmond, W. L. Langham, and T. T. Trujillo, "Comparative Metabolism of Tritiated Water by Mammals," J. Cellular Comp. Physiol. 59, 45 (1962).
18. E. P. Radford, T. Merz, C. Ray, R. M. Herriott, C. W. Kruse, C. H. Southwick, and B. K. Sladen, "Statement of Concern," Environment 11 (7), (20 September, 1969).
19. T. E. Smith and R. T. Taylor, Incorporation of Tritium from Tritiated Water into Carbohydrates, Lipids, and Nucleic Acids, Lawrence Radiation Laboratory, Livermore, Rept. UCRL-50781 (1969).
20. H. P. Metzger, M. Cuenod, A. Grynbaum, and H. Waelsch, "The Use of Tritium Oxide as a Biosynthetic Precursor of Macromolecules in Brain and Liver," J. Neurochem. 14, 99 (1967).
21. J. R. Hinrichs, R. O. Petersen, and R. Baserga, "Incorporation of Thymidine into DNA of Mouse Organs," Arch. Pathol. 78, 245 (1964).
22. J. R. Rubini, E. P. Cronkite, V. P. Bond, and T. M. Fliedner, "The Metabolism and Fate of Tritiated Thymidine in Man," J. Clin. Invest. 39, 909 (1960).
23. B. E. Lambert and R. J. Clifton, "Radiation Doses Resulting from the Ingestion of Tritiated Thymidine by the Rat," Health Phys. 15, 3 (1968).
24. W. R. Guild, "Hazards from Isotopic Tracers," Science 128, 1308 (1958).
25. F. T. Hatch, J. A. Mazrimas, G. G. Greenway, J. J. Koranda, and J. L. Moore, Studies on Liver DNA in Tritiated Kangaroo Rats Living at Sedan Crater, Lawrence Radiation Laboratory, Livermore, Rept. UCRL-50461 (1968).
26. R. Moore, "A Comparison of HTO in Plasma and Expired Water Vapor," Health Phys. 7, 161 (1962).
27. J. H. M. Bowen, Trace Elements in Biochemistry (Academic Press, New York, 1966).
28. V. P. Bond and L. E. Feinendegen, "Intranuclear ^{3}H Thymidine: Dosimetric, Radiobiological and Radiation Protection Aspects," Health Phys. 12, 1007 (1966).
29. G. Marin and M. A. Bender, "A Comparison of Mammalian Cell-Killing by Incorporated ^{3}H-Thymidine and ^{3}H-Uridine," Intern. J. Radiation Biol. 7, 235 (1963).
30. G. Marin and M. A. Bender, "Survival Kinetics of HeLa S-3 Cells after Incorporation of ^{3}H-Thymidine or ^{3}H-Uridine," Intern. J. Radiation Biol. 7, 221 (1963).
31. R. M. Drew and R. B. Painter, "Action of Tritiated Thymidine on the Clonal Growth of Mammalian Cells," Radiation Res. 11, 535 (1959).
32. R. M. Drew and R. B. Painter, "Further Studies on the Clonal Growth of HeLa S-3 Cells Treated with Tritiated Thymidine," Radiation Res. 16, 303 (1962).
33. R. B. Painter, R. M. Drew, and B. G. Giaugue, "Further Studies on Deoxyribonucleic Acid Metabolism in Mammalian Cell Cultures," Exptl. Cell Res. 21, 98 (1960).

34. B. E. Lambert, "Cytological Damage Produced in the Mouse Testes by Tritiated Thymidine, Tritiated Water and X-Rays," Health Phys. 17, 547 (1969).
35. H. A. Johnson and E. P. Cronkite, "The Effect of Tritiated Thymidine on Mouse Spermatogonia," Radiation Res. 11, 825 (1959).
36. W. C. Dewey, R. M. Humphrey, and B. A. Jones, "Comparisons of Tritiated Thymidine, Tritiated Water and Cobalt-60 Gamma Rays in Inducing Chromosomal Aberrations," Radiation Res. 24, 214 (1965).
37. M. A. Bender, P. C. Gooch, and D. M. Prescott, "Aberrations Induced in Human Luekocyte Chromosomes by ^{3}H-Labeled Nucleosides," Cytogenetics 1, 65 (1962).
38. R. Baserga, H. Lisco, and W. E. Kisieleski, "Tumor Induction in Mice by Radioactive Thymidine," Radiation Res. 29, 583 (1966).
39. A. J. Bateman and A. C. Chandley, "Mutations Induced in the Mouse with Tritiated Thymidine," Nature 193, 705 (1962).
40. R. C. Greulich, "Deleterious Influence of Orally Administered Tritiated Thymidine on Reproductive Capacity of Mice," Radiation Res. 14, 83 (1961).
41. W. D. Kaplan, H. D. Gugler, K. K. Kidd, and V. E. Tinderholt, "Nonrandom Distribution of Lethals Induced by Tritiated Thymidine in Drosophila melanogaster," Genetics 49, 701 (1964).
42. W. D. Kaplan, H. D. Gugler, and K. K. Kidd, "Distribution of Lethals Induced by Tritiated DNA Precursors in Drosophila melanogaster," Genetics 53, 499 (1966).
43. M. Rachmeler and A. B. Pardee, "Loss of Viability and β-Galactosidase-Forming Ability as a Consequence of Tritium Decay in Escherichia coli," Biochim. Biophys. Acta 68, 62 (1963).
44. A. A. Khan and J. E. Wilson, "Studies of Turnover in Mammalian Subcellular Particles: Brain Nuclei, Mitochondria and Microsomes," J. Neurochem. 12, 81 (1965).
45. International Commission on Radiological Protection, The Evaluation of Risks from Radiation, ICRP Publ. 8 (Pergamon Press, New York, 1966).
46. R. W. Miller, "Delayed Radiation Effects in Atomic-Bomb Survivors," Science 166, 569 (1969).
47. A. Holzer, "Gasbuggy in Perspective," in Proc. Am. Nucl. Soc. Symposium on Engineering with Nuclear Explosives, January 14-16, 1970, Las Vegas, Nevada (in press).
48. E. H. Fleming, Lawrence Radiation Laboratory, Livermore, private communication (1970).
49. J. B. Knox, Lawrence Radiation Laboratory, Livermore, private communication (1959).
50. Y. C. Ng, C. A. Burton, S. E. Thompson, R. K. Tandy, H. K. Kretner, and M. W. Pratt, Prediction of the Maximum Dosage to Man from the Fallout of Nuclear Devices. IV. Handbook for Estimating the Maximum Internal Dose from Radionuclides Released to the Biosphere, Lawrence Radiation Laboratory, Livermore, Rept. UCRL-50163 Part IV (1968).

51. C. A. Burton, A Graphical Method for Calculating the Dose in Man from Individual Radionuclides in Fallout, Lawrence Radiation Laboratory, Livermore, Rept. UCRL-14922 (1966).
52. J. J. Koranda and J. R. Martin, "The Persistence of Radionuclides at Sites of Nuclear Detonations," in Biological Implications of the Nuclear Age, Proc. Bio-Medical Symposium, Lawrence Radiation Laboratory, Livermore, March 5-7, 1969, CONF-690303, p. 159 (1969).
53. J. R. Martin, C. F. Jordan, J. J. Koranda and J. R. Kline, "Radioecological Studies of Tritium Movement in a Tropical Rain Forest," in Proc. Am. Nuclear Soc. Symposium on Engineering with Nuclear Explosives, January 14-16, 1970, Las Vegas, Nevada (in press).
54. E. Pinson and W. H. Langham, "Physiology and Toxicology of Tritium in Man," J. Appl. Physiol. 10, 108 (1957).
55. W. L. Russell, "The Effect of Radiation Dose Rate and Fractionation on Mutation in Mice," in Repair from Genetic Radiation Damage, F. H. Sobels, Ed. (Pergamon Press, Oxford, 1963).
56. W. L. Russell, "Recent Studies on the Genetic Effects of Radiation in Mice," Pediatrics 41, 223 (1968).
57. M. M. Elkind, C. Kamper, W. B. Moses, and H. Sutton-Gilbert, "Sublethal-Lethal Radiation Damage and Repair in Mammalian Cells," Recovery and Repair Mechanisms in Radiobiology, Brookhaven National Laboratory Symposium, Brookhaven National Laboratory, Upton, N. Y. Rept. BNL-50058 (1967), p. 134.
58. S. Wolff, "Some Postirradiation Phenomena that Affect the Induction of Chromosome Aberrations," J. Cellular Comp. Physiol. (Suppl. 1) 151 (1961).
59. S. Wolff, "On the Chemistry of Chromosome Continuity," Natl. Cancer Inst. Monogr. 18, 155 (1965).
60. J. E. Cleaver and R. B. Painter, "Evidence for Repair Replication of HeLa Cell DNA Damaged by Ultraviolet Light," Biochim. Biophys. Acta 161, 552 (1968).
61. R. E. Rasmussen and R. B. Painter, "Radiation-Stimulated DNA Synthesis in Cultured Mammalian Cells," J. Cell Biol. 29, 11 (1966).
62. J. E. Cleaver, "Repair Replication of Mammalian Cell DNA: Effects of Compounds that Inhibit DNA Synthesis in Dark Repair," Radiation Res. 37, 334 (1969).
63. R. B. Painter and J. E. Cleaver, "Repair Replication, Unscheduled DNA Synthesis, and the Repair of Mammalian DNA," Radiation Res. 37, 451 (1969).
64. R. B. Painter and J. E. Cleaver, "Repair Replication in HeLa Cells after Large Doses of X-irradiation," Nature 216, 369 (1967).
65. J. T. Lett, I. Caldwell, C. J. Dean, and P. Alexander, "Rejoining of X-Ray Induced Breaks in the DNA of Leukemia Cells," Nature 214, 790 (1967).
66. B. Weiss and C. C. Richardson, "Enzymatic Breakage and Joining of Deoxyribonucleic Acid, I. Repair of Single-Strand Breaks in DNA by an Enzyme System from Escherichia coli Infected with T4 Bacteriophage," Proc. Natl. Acad. Sci. U.S. 57, 1021 (1967).

67. S. B. Zimmerman, J. W. Little, C. K. Oshinsky, and M. Gellert, "Enzymatic Joining of DNA Strands: A Novel Reaction of Diphosphopyridine Nucleotide," Proc. Natl. Acad. Sci. U.S. 57, 1841 (1967).
68. R. B. Kelly, M. R. Atkinson, J. A. Huberman, and A. Kornberg, "Excision of Thymine Dimers and Other Mismatched Sequences by DNA Polymerase of Escherichia coli," Nature 224, 495 (1969).
69. L. R. Anspaugh and W. L. Robison, Quantitative Evaluation of the Biological Hazards of Radiation Associated with Project Ketch, Lawrence Radiation Laboratory, Livermore, Rept. UCID-15325 (1968).
70. A. C. Andersen and L. S. Rosenblatt, "The Effect of Whole-Body X-Irradiation on the Median Lifespan of Female Dogs (Beagles)," Radiation Res. 39, 177 (1969).
71. J. B. Storer, "Late Effects: Extrapolation to Low Dose Rate Exposures," Health Phys. 17, 3 (1969).
72. J. W. Gofman and A. R. Tamplin, "Low Dose Radiation, Chromosomes, and Cancer," in Proc. 16th IEEE Nuclear Science Symposium, IEEE Trans. Nuclear Sci. (in press).

Appendix
Quantitative Evaluation of Risk Due to Exposure to Low Levels of Whole-Body Radiation

INTRODUCTION

The purpose of this appendix is to provide quantitative estimates of the detrimental effects to human populations resulting from exposure to ionizing radiations. Such estimates should be of value in objectively assessing the risks versus benefits of proposed projects involving the exposure of human populations to radiation.

It has been known for many years that exposure to sublethal levels of radiation can produce genetic mutations and malignant diseases such as leukemia, bone sarcoma, thyroid carcinoma, etc. There still remains, however, considerable argument concerning the dose-response relationships, that is, whether the number of effects of a given type is directly proportional to the dose, or whether there is a dose threshold below which no effects occur. This question is extremely important from the standpoint of public safety because the doses of concern are generally extremely small and frequently below the level of normal background radiation of approximately 100 mrad per year.[1] Our knowledge concerning dose-effect relationships is almost entirely confined, however, to dose regions above 100 rad, which is 1000 times greater than background. Only in the case of genetic mutations can a strong argument be made that the response is linear over wide ranges of doses and at dose levels down to a few rad.[2,3] A second very significant question is that of dose-rate effects. If there are recovery processes operating to repair radiation damage, the period of time over which exposure is delivered becomes a very important variable. Unfortunately, most of our knowledge about radiation effects concerns doses delivered over short periods of time, and it is again only for genetic effects that we can be reasonably certain of the possible magnitude of this effect.[2,3]

Adequate data concerning the dose-rate relationships for the production of malignant diseases do not exist for low doses because the effects are so small they could not be observed except by analyzing impossibly large population groups. Effects in humans at low doses have been seen only for the special cases of irradiation of embryos[4,5] and leukemia induction in adults,[6] but even here no data exist for the low dose rates of most significance to this evaluation.

Thus, while the dose-response and dose-rate relationships are of utmost importance and of considerable scientific interest, it is necessary to assume some kind of dose-response, dose-rate relationship to make a quantitative estimate of the risks involved for exposure to low doses of radiation delivered at low dose rates. In general, the choice must always be made that the specific risk increases linearly with dose, and that effects do not vary with changes in dose

rate. This approach is taken not only because it is the simplest one, but because it is also the most conservative in terms of public safety.

Throughout the following considerations the assumption is also made that the type of exposure of concern is one that would deliver whole-body doses. Considerably different approaches would be called for if, for example, the main concern were the inhalation of particulate radioactive matter, or the ingestion of radioactive iodine.

It should be pointed out that many national and international committees have been charged with the evaluation of the risks considered here, and many excellent discussions of the problem have been published.[1,7-11] The results calculated here are at least as conservative as those recommended by the International Commission on Radiological Protection (ICRP),[7] although in some cases quantitative estimates have been made for risks for which the ICRP feels adequate data may not exist to justify such calculations. In such cases our prime consideration has not been one of scientific exactness but one of extreme caution. In other words, we have chosen to err on the side of overestimating the risk where adequate data and scientific consensus are not available.

The calculations that follow clearly demonstrate that the risk of genetic death is the most important numerically, especially if the effect over all subsequent generations is considered. Genetic death in this sense, however, refers primarily to effects that would result in fetal death or reduced fertility.

ESTIMATION OF GENETIC RISKS

In evaluating the hazards of radiation-induced genetic effects on a population, one should first know the magnitude of the genetic effects occurring naturally in that population. The effects due to radiation exposure can then be compared to the natural or spontaneous genetic effects occurring in the population. The natural frequency of gene mutations which lead to genetic death* will be estimated using three different methods. The frequency of radiation-induced mutations will be determined from an analysis of the available data, and the genetic detriment to all subsequent generations and to the first generation alone will be determined.

Calculation of the Natural Mutation Frequency

The natural gene mutation frequency for specific types of mutations (i.e., recessive lethal, recessive visible, dominant visible) can be estimated in human populations by several methods. Different approaches for estimating the recessive lethal mutation frequency over the X-chromosone give a result of approximately 0.003 mutations per X-chromosome per generation.[11,14,15] If one assumes that the X-chromosome is 5% of the total mutable genome (genome is the total chromosome complement of a germ cell, i.e., sperm or egg) and when account is taken of selection against recessive

*Genetic death refers to the eventual extinction of a deleterious gene. This would be evidenced by abortion, stillbirth, prereproductive death, early embryonic death, lowered fertility or sterility.[12,13]

lethals, extrapolation over the whole genome gives a spontaneous mutation frequency of 0.14 mutations per germ cell per generation.[11]

Estimates of total mutation frequency can also be derived from the frequency of occurrence of dominant, sex-linked, and recessive visible conditions. Mutation frequencies of the causal genes range from 0.2 to 20 per million per generation.[7,16] A reasonable number for calculation would be 5 mutations per million genes per generation.[7,11,12] If one then takes the number of genes in man to be 20,000,[7,11,17] one can calculate a total spontaneous mutation rate of 0.1 per germ cell per generation:

$$\frac{5 \times 10^{-6} \text{ mutations}}{\text{gene-generation}} \times \frac{2 \times 10^{4} \text{ genes}}{\text{germ cell}}$$

$$= \frac{0.1 \text{ mutations}}{\text{germ cell-generation}}$$

These are the mutations that would lead to genetic death.

If from the above we accept a total mutation frequency per germ cell of 0.1, we can calculate that the total new mutations per generation per million people is 200,000:

$$\frac{0.1 \text{ mutations}}{\text{germ cell-generation}} \times \frac{2 \text{ germ cells}}{\text{person}}$$

$$\times 10^{6} \text{ persons} = \frac{200,000 \text{ mutations}}{\text{generation}}$$

At genetic equilibrium the number of new mutations produced per generation is balanced by the same number of eliminations. Therefore 20% or 200,000 per million are eliminated per generation from natural mutation.

The natural occurrence of mutations leading to genetic death can also be arrived at by a third method.[12,18] With data from inbred populations, the number of detrimental genes carried by each person and their selective disadvantage can be estimated and used to determine the total number of eliminations per generation. The ICRP report[7] assumes that an individual carries 8 small dominant genes, each with an independent risk of elimination of 0.025 per generation. The total chance of elimination per generation is then 8×0.025 or 0.2 per person.

Calculation of the Radiation-Induced Mutation Frequency

The values that must be used to calculate the frequency of mutation induction from radiation come almost exclusively from animal data, with only indirect evidence from human data. Data from experiments on mice serve as the primary source of information, and a reasonable estimate from these data for the induced mutation frequency is 10^{-7} mutations per gene per rad per germ cell.[3,19] This value represents data from acute exposure experiments. Dose-rate effects have been shown to exist in spermatogonia and oogonia in mice[3,19]; a chronic exposure has about one-fourth the mutational effect of the same dose given in an acute exposure. In the following calculations, however, we will use the value from acute-exposure experiments, since it will provide an upper estimate of the effects.

The value of 10^{-7} mutations per gene per rad per germ cell is supported

indirectly as a reasonable estimate for humans from data collected by Neel and Schull[20] from Hiroshima and Nagasaki. They found no increase in the genetic deaths of children from irradiated parents above that of the control population. This does not mean that no additional mutations occurred from these doses but that the estimate of the mutation frequency for humans could not be much higher than that determined from mouse data, or significant results would have been seen.[8,12,17] Using these numbers one finds a mutation frequency of 2000 mutations per one million germ cells per rad or 4000 mutations per one million persons per rad:

$$\frac{10^{-7}\ \text{mutations}}{\text{gene-rad}} \times 2 \times 10^{4}\ \frac{\text{genes}}{\text{germ cell}} \times 2\ \frac{\text{germ cells}}{\text{person}} \times 10^{6}\ \text{persons} \times 1\ \text{rad} = 4000\ \text{mutations}$$

or genetic deaths over all generations. This number turns out to be quite comparable to the overall radiation-induced mutation frequency determined for mice from different studies.[21,22,23,24]

Even though these calculations were made on a conservative basis the ICRP[7] uses a value given by Muller of 3600 mutations per one million germ cells per rad or 7200 genetic deaths per million persons per rad over all generations. Since this is an even higher estimate of the risk we will accept it as our final value for the number of genetic deaths over all generations resulting from the exposure of one million people to one rad.

If 2.5% of these mutations are expressed or eliminated in the first generation,[7] then the risk in the first generation is 180 genetic deaths per million persons per rad:

$$3.6 \times 10^{-3}\ \frac{\text{mutations}}{\text{germ cell-rad}} \times 2\ \frac{\text{germ cells}}{\text{person}} \times 10^{6}\ \text{persons} \times 1\ \text{rad} \times 0.025 = 180\ \text{genetic deaths}$$

in the first generation. This is comparable to the 200,000 genetic deaths in this generation due to the occurrence of natural spontaneous mutations.

The 180 genetic deaths in the first generation due to radiation exposure of 1 rad can also be compared to the number of genetic deaths out of the total which are expressed in the first generation from mutations within the parental generation only. This number is 5000 genetic deaths[7]:

$$0.1\ \frac{\text{mutations}}{\text{germ cell}} \times 2\ \frac{\text{germ cells}}{\text{person}} \times 10^{6}\ \text{persons} \times 0.025 = 5000\ \text{genetic deaths}$$

from parental generation.

The estimate of the genetic risk resulting from exposure of a population to radiation can also be determined by a different method used by Crow et al.[18,25,26] The values for total generation risk and first-generation risk obtained from this method are essentially the same as those calculated above.

MALIGNANT DISEASES

That exposure to radiation could lead to the production of malignant diseases

was one of the earliest observations of human effects. Of all the malignant diseases, leukemia is particularly easy to observe and this disease will be considered separately. All other malignant diseases will be considered as a group.

Leukemia

Data pertaining to the production of malignant diseases are certainly the most complete for leukemia. This is true for two reasons: (1) the types of leukemia produced by radiation (acute leukemia and chronic granulocytic leukemia) are practically synonymous with death and thus are dramatic and easily traced events, and (2) the period of time between radiation exposure and the appearance of malignant disease is shortest for leukemia.

Risk of Developing Leukemia Following the Exposure of Adults

Several sets of human data are available to assess this risk. The largest population at risk is represented by the survivors of the atomic bomb blasts at Hiroshima and Nagasaki. Brill et al.[27] reported on 149 confirmed leukemia cases arising from 2,116,478 man years at risk up to 1958. An attempt was made to provide proper dose estimates for 76 of these cases and thus to calculate the dose-response factors. The best possible estimate for both cities was 1 to 2 cases of leukemia per million people per year at risk per rad, averaged over the 14 years covered by the study. This estimate is compatible with a linear dose response between 100 and 500 rad. New cases were observed to begin appearing within one and a half to two years after exposure; maximum risk occurred at four to seven years after exposure and then fell off considerably, although the increased risk was still apparent 14 years after the exposure.

The second large group of individuals studied comprised 11,287 male patients suffering from ankylosing spondylitis who were treated by therapeutic radiation to the spine. Court-Brown and Doll[28] subsequently followed up this group and found 37 cases of leukemia arising between one and twenty years following exposure. Court-Brown[29] calculated that the probability of developing leukemia per year lies between 1 and 2 per million people at risk per roentgen (R) averaged over the whole red marrow system. The latent period for the onset of these cases appears to reach a peak during the fourth and fifth years and then to fall off.[30,31]

A large number of other studies have also demonstrated a correlation between radiation exposure and an increased incidence of leukemia (see Ref. 10 for a summary). Most of them, however, provide little additional dose-response information either because the number of cases are small and/or because the doses are not precisely known. Lewis[32] and Hempelmann[33] have also calculated dose-response estimates from data concerning the increased incidence of leukemia among U.S. radiologists and among children exposed to therapeutic radiation. All values are essentially consistent with a risk of 1 to 2 cases of leukemia per million persons at risk per rad per year.

From an evaluation of these data, the ICRP has stated that the total leukemia risk would appear to be of the order of 20 cases per million persons (over a

lifetime) per rad.[7] This assumes a linear relationship and a declining risk after a maximum risk period. This is contrasted with the natural incidence of leukemia of 5000 per million persons over a lifetime.[34,35]

Risk of Developing Leukemia after Exposure in utero

Several studies have demonstrated that in utero irradiation is followed by an increased incidence of leukemia. The largest studies and those with the necessary follow-up periods are those of Stewart, Webb, and Hewitt[4] and MacMahon.[5] Other studies are reviewed in the United Nations report[10] and in MacMahon's paper.[5] According to MacMahon's analysis the best estimate of the relative risk involved is 1.40,[5,36] that is, those individuals receiving whole body radiation in utero have a 40% greater chance of developing leukemia before the age of 10. MacMahon further states that the excess risk of developing leukemia before the age of 10 is 18 per 100,000 live births. Assuming the dose associated with these exposures to be 2 R (Ref. 36), the risk is then 9 per million persons per rad per year. Hempelmann's calculation of this value[33] is 10 to 11 cases of leukemia per million persons per rad per year. Sternglass's analysis of these data,[37] however, is that 1.7 R to the fetus would double the natural incidence of 46 cases of leukemia before the age of 10 per 100,000 live births which is equivalent to a risk of 26 cases per million persons irradiated in utero per rad per year.

Most studies have shown that the incidence of excess leukemia reaches a maximum at age 5 through age 7 and is exhausted soon after. Thus, assuming the most conservative estimate of the risk and assuming that the risk is exhausted after 10 years, the risk of developing leukemia following in utero irradiation is 260 per million live births per rad. The natural incidence is reported to be 460 per million live births.[5]

Malignant Diseases Other than Leukemia

In considering only the effects of whole-body radiation, the induction of specific cancer types need not be considered. It is sufficient to consider the total number of malignancies other than leukemia that may be produced.[7]

Risk of Developing Malignant Diseases Other than Leukemia Following the Exposure of Adults

One source of data for evaluating this risk is again the Japanese survivors of the atomic bomb blasts. Here it has been noted that leukemia contributes about 50% of the total mortality from malignant diseases 5 to 15 years after the exposure.[38]

The spondylitics treated with radiation show a somewhat similar pattern; the mortality due to cancer of heavily irradiated sites is 1.5 times that due to leukemia alone.[30] This is observed with an average followup period of 13 years.

However, deaths due to other malignant diseases, unlike leukemia, do not show any clear peak incidence and if anything, the incidence appears to be increasing with years past exposure.[39]

The ICRP has taken as its working hypothesis that the total increase in cancer mortality (other than that from leukemia) within 20 years after exposure, is

equal to the mortality of leukemia alone.[7] It seems reasonable to double this figure for a lifetime after exposure on the basis of the later data from the spondylitics.[30] Thus if the incidence should keep on increasing indefinitely the risk may be underestimated, but on the other hand most of the malignant diseases are not nearly so fatal as leukemia.

The risk for all malignant diseases except leukemia would then be 40 cases per million persons per rad. The natural incidence of mortality from malignant diseases other than leukemia is at least 20 times the incidence from leukemia or 100,000 cases per million people per lifetime.[35]

Risk of Developing Malignant Diseases Other than Leukemia after Exposure in utero

Both MacMahon[5] and Stewart et al.[4] found that following irradiation in utero, the incidence of all types of malignant diseases was increased in essentially the same proportion as that for leukemia. However, no group of children has been followed longer than 10 years, and there are no adequate data to support the idea that deaths from these malignant diseases other than leukemia have peaked out in the first 10 years. Therefore, the same assumption will be made that over a life-span, death is twice as probable from other malignant diseases than it is for leukemia, or that the risk is 520 per million live births per rad. The associated natural incidence is 460 per million live births before the age of ten,[5] and approximately 100,000 per million live births considered over a lifetime.

NONSPECIFIC LIFE-SPAN SHORTENING

Following the exposure of a large population of animals to radiation, an increased mortality rate is frequently noted. This effect is not accounted for by the increased frequency of any single disease process, but the effect is as though all animals had suddenly become "older" in a physiological sense. The risk factor in terms of collective lives lost per million persons at risk is calculated below..

Data from Animal Experiments

Animal data on the shortening of life-span induced by radiation show a considerable variation of values. Failla and McClement, with mathematical methods to extrapolate mouse data to man, concluded that approximately 0.5% of a normal life-span would be lost due to an individual's exposure to 100 rad.[40] Russell, however, found a 7.8% life-span reduction following 100 rad of neutron radiation to mice.[41] Other investigators have reported values of between 2 and 5% life-span reduction for mice irradiated acutely with x rays or gamma rays.[42-46] Storer et al. found a life-span reduction of 5 to 7% for mice per 100 rads of neutron irradiation.[44] Lorenz et al.[47] investigated two different strains of mice and guinea pigs irradiated at several dose rates. At exposure levels sufficiently low to avoid acute mortality, nonspecific life-span shortening effects were found to be between 1 and 3% per 100 R of exposure. Blair[48] has reviewed the data on these effects in mice and rats and has reported a value of between 1 and 2% reduction in life-span per 100 R as being a reasonable estimate.

Russell[41] has also reported that the offspring of irradiated mice suffer nonspecific life-span reduction, but the much larger and comprehensive study of Spalding et al.[46] has failed to demonstrate such effects.

Experimental Data from Human Populations

Court-Brown and Doll[49] have found no evidence of a nonspecific life-span shortening in their study of British radiologists, some of whom were believed to have been exposed to excessively high levels of radiation. On the other hand, Seltzer and Sartwell[50] have found a decrease in life expectancy among American radiologists when they are compared to other medical specialists. The reduction in life-span appears to be of the order of 4 to 5%. Assuming that the total dose received was 100 rad, the risk factor would be a 4 to 5% reduction in life-span per 100 rad. The actual doses received by the radiologists are, however, unknown and were certainly highly variable among individual radiologists. Various estimates for the average dose to this group range from 100 rad[32] to 500 rad.[51] Studies on the survivors of the atomic bomb blasts in Japan are plagued with many difficulties, and as yet no general conclusions appear to be warranted.[38]

Quantitative Estimate of the Nonspecific Life-Shortening Risk

A reasonable approximation of the upper limit of the risk, based on data from both man and animal experiments, is a 7% reduction in life-span per 100 rad of exposure. Assuming a linear extrapolation down to low dose levels, the collective risk would be 700 lives lost per million persons exposed per rad.

References

1. B. Lindell and R. L. Dobson, Ionizing Radiation and Health, World Health Organization, Public Health Paper No. 6 (Geneva, 1961).
2. W. P. Spencer and C. Stern, "Experiments to Test the Validity of the Linear r Dose. Mutation Frequency Relation in Drosophila at Low Dosage," Genetics 33, 43 (1948).
3. W. L. Russell, "The Effect of Radiation Dose Rate and Fractionation on Mutation in Mice," in Repair from Genetic Radiation Damage, F. H. Sobels, Ed. (Pergamon Press, Oxford, 1963).
4. A. Stewart, J. Webb, and D. Hewitt, "A Survey of Childhood Malignancies," Brit. Med. J. 1, 1495 (1958).
5. B. MacMahon, "Prenatal X-Ray Exposure and Childhood Cancer," J. Natl. Cancer Inst. 28, 1173 (1962).
6. A. Stewart, W. Pennybacker, and R. Barker, "Adult Leukemias and Diagnostic X-Rays," Brit. Med. J. 2, 882 (1962).
7. International Commission on Radiological Protection, The Evaluation of Risks from Radiation, ICRP Publ. 8 (Pergamon Press, New York, 1966).
8. The Hazards to Man of Nuclear and Allied Radiations. A Second Report to the Medical Research Council (Gt. Brit.), (H. M. Stationery Office, London, 1960).
9. Committee on the Biological Effects of Atomic Radiation, The Biological Effects of Atomic Radiation, National Academy of Sciences—National Research Council Summary Reports, Washington, D. C. (1960).
10. Report of the United Nations Scientific Committee on the Effects of Atomic Radiation, General Assembly Official Records: Nineteenth Session, Suppl. 14 (A/5814), (United Nations, New York, 1964).
11. Report of the United Nations Scientific Committee on the Effects of Atomic Radiation, General Assembly Official Records: Twenty-First Session, Suppl. 14 (A/6314), (United Nations, New York, 1966).
12. J. F. Crow, "The Estimation of Spontaneous and Radiation-Induced Mutation Rates in Man," Eugenics Quart. 3, 201 (1956).
13. H. J. Muller, "Our Load of Mutations," Human Genetics 2, 111 (1950).
14. O. Frata-Pessoa, "On the Number of Gene Loci and the Total Mutation Rate in Man," Am. Naturalist 95, 217 (1961).
15. J. Lejeune and R. Turpin, "Mutations Radio Induites Chez l'Humme et Dose de Doublement. Sur la Validité d'une Estimation Directe," Compt. Rend. 244, 2425 (1957).
16. L. S. Penrose, "The Spontaneous Mutation Rate in Man," Appendix C in The Hazards to Man of Nuclear and Allied Radiations (H. M. Stationery Office, London, 1956), p. 90.
17. C. Stern, "Radiation and Population Genetics," Chapter 9 in Radiation Biology and Medicine, W. D. Claus, Ed., (Addison-Wesley, Reading, 1958), p. 206.

18. J. F. Crow, "Possible Consequences of an Increased Mutation Rate," Eugenics Quart. 4, 67 (1957).
19. M. F. Lyon and T. M. Morris, "Mutation Rates at a New Set of Specific Loci in the Mouse," Genetics Res. 7, 12 (1966).
20. J. V. Neel and W. J. Schull, The Effect of Exposure to the Atomic Bombs on Pregnancy Termination in Hiroshima and Nagasaki, Pub. 461 (National Academy of Sciences, National Research Council, Washington, D. C., 1956).
21. M. F. Lyon, R. J. S. Phillips, and A. G. Searle, "The Overall Rates of Dominant and Recessive Lethal and Visible Mutation Induced by Spermatogonial X-Irradiation of Mice," Genetics Res. 5, 448 (1964).
22. K. G. Lüning, "Studies of Irradiated Mouse Populations. III. Accumulation of Recessive Lethals," Mutation Res. 1, 86 (1964).
23. R. J. S. Phillips, "A Comparison of Mutation Induced by Acute X and Chronic Gamma Irradiation in Mice," Brit. J. Radiol. 34, 261 (1961).
24. T. Sugahara, "Genetic Effects of Chronic Irradiation Given to Mice Through Three Successive Generations," Genetics 50, 1143 (1964).
25. J. F. Crow, in Hearings before the Special Subcommittee on Radiation, Joint Committee on Atomic Energy, Congress of the United States, Part 2, (U.S. Government Printing Office, Washington, D. C., 1957), p. 1009.
26. N. E. Morton, J. F. Crow, and H. J. Muller, "An Estimate of the Mutational Damage in Man from Data on Consanguinous Marriages," Proc. Natl. Acad. Sci. U.S. 42, 855 (1956).
27. A. B. Brill, M. Tomonaga, and R. M. Heyssel, "Leukemia in Man Following Exposure to Ionizing Radiation. A Summary of the Findings in Hiroshima and Nagasaki, and a Comparison with Other Human Experience," Ann. Internal Med. 56, 590 (1962).
28. W. M. Court-Brown and R. Doll, "Leukaemia and Aplastic Anaemia in Patients Treated with X-Rays for Ankylosing Spondylitis," Appendix B in The Hazards to Man of Nuclear and Allied Radiations (H. M. Stationery Office, London, 1956), p. 87.
29. W. M. Court-Brown, "Radiation-Induced Leukemia in Man, with Particular Reference to the Dose-Response Relationship," J. Chronic Diseases 8, 113 (1958).
30. W. M. Court-Brown, "Mortality from Cancer and Other Causes after Radiotherapy for Ankylosing Spondylitis," Brit. Med. J. 2, 1327 (1965).
31. M. E. Wise, "The Latent Period and its Variation in Human Leukaemia Induced by X-Rays," Health Phys. 4, 250 (1961).
32. E. B. Lewis, "Leukemia and Ionizing Radiation," Science 125, 965 (1957).
33. L. H. Hempelmann, "Epidemiological Studies of Leukemia in Persons Exposed to Ionizing Radiation," Cancer Res. 20, 18 (1960).
34. H. L. Friedell, in Hearings before the Special Subcommittee on Research, Development, and Radiation, Joint Committee on Atomic Energy, Congress of the United States, Part 2 (U.S. Government Printing Office, Washington, D. C., 1963), p. 581.

35. Vital Statistics of the United States, 1965, Part II. Mortality, National Center for Health Statistics, U.S. Department of Health, Education, and Welfare, (U.S. Government Printing Office, Washington, D.C., 1967).
36. B. MacMahon, in Hearings before the Special Subcommittee on Research, Development, and Radiation, Joint Committee on Atomic Energy, Congress of the United States, Part 2 (U.S. Government Printing Office, Washington, D.C., 1963), p. 594.
37. E. J. Sternglass, "Cancer: Relation of Prenatal Radiation to Development of the Disease in Childhood," Science 140, 1102 (1963).
38. S. Jablon, M. Ishida, and M. Yamasaki, "Studies of the Mortality of A-Bomb Survivors. 3. Description of the Sample and Mortality, 1950-1960," Radiation Res. 25, 25 (1965).
39. J. W. Pifer, E. T. Toyooka, R. W. Murray, W. R. Ames, and L. H. Hempelmann, "Neoplasia in Children Treated with X Rays for Thymic Enlargement. I. Neoplasms and Mortality," J. Natl. Cancer Inst. 31, 1333 (1963).
40. G. Failla and P. McClement, "The Shortening of Life by Chronic Whole-Body Irradiation," Am. J. Roentgenol. Radium Therapy Nucl. Med. 78, 946 (1957).
41. W. L. Russell, "Shortening of Life in the Offspring of Male Mice Exposed to Neutron Radiation from an Atomic Bomb," Proc. Natl. Acad. Sci. U.S. 43, 324 (1957).
42. R. F. Kallman and H. I. Kohn, "Life-Shortening by Whole- and Partial-Body X-Irradiation in Mice," Science 128, 301 (1958).
43. P. Alexander, and O. Connell, "Shortening of the Life Span of Mice by Irradiation with X-Rays and Treatment with Radiomimetic Chemicals," Radiation Res. 12, 38 (1960).
44. J. B. Storer, B. S. Robers, I. U. Boone, and P. S. Harris, "Relative Effectiveness of Neutrons for Production of Delayed Biological Effects. II. Effect of Single Doses of Neutrons from an Atomic Weapon on Life Span of Mice," Radiation Res. 8, 71 (1958).
45. P. J. Lindop and J. Rotblat, "Shortening of Lifespan of Mice as a Function of Age at Irradiation," Gerontologia 3, 122 (1959).
46. J. F. Spalding, M. Brooks, and P. McWilliams, "Observations on Lifespan, Radioresistance, and Productivity in Offspring from 5 to 25 Generations of X-Irradiated Male Mice," Genetics 50, 1179 (1964).
47. E. Lorenz, L. O. Jacobson, W. E. Heston, M. Shimkin, A. B. Eschenbrenner, M. K. Deringer, J. Doniger, and R. Schweisthal, Chapter 3 in "Effects of Long-Continued Total-Body Gamma Irradiation on Mice, Guinea Pigs, and Rabbits. III. Effects on Life Span, Weight, Blood Picture, and Carcinogenesis and the Role of the Intensity of Radiation," Biological Effects of External X and Gamma Radiation (R. E. Zirkle, Ed.), Part I (McGraw-Hill, New York, 1954), p. 24.
48. H. A. Blair, Data Pertaining to Shortening of Life Span by Ionizing Radiation, Rochester Univer., Rochester, N. Y., Report, UR-442 (May 1956).

49. W. M. Court-Brown and R. Doll, "Expectation of Life and Mortality from Cancer among British Radiologists," Brit. Med. J. 2, 181 (1958).

50. R. Seltzer and P. E. Sartwell, "The Influence of Occupational Exposure to Radiation on the Mortality of American Radiologists and Other Medical Specialists," Am. J. Epidemiol. 81, 2 (1965).

51. H. B. Jones, in Hearings before the Special Subcommittee on Radiation, Joint Committee on Atomic Energy, Congress of the United States, Part 2 (U. S. Government Printing Office, Washington, D. C., 1957), p. 1116.

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